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EVALUATION OF SCOPE FOR HARM FOR WHITE SHARK (*CARCHARODON CARCHARIAS*) IN ATLANTIC CANADA

Context

In Canada, White Shark (*Carcharodon carcharias*) was assessed as Endangered in 2006 by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC 2006) and was listed on Schedule 1 of *Species at Risk Act* (SARA) in 2011. A Recovery Potential Assessment conducted by Fisheries and Oceans Canada (DFO) prior to listing concluded that there was no scope for allowable harm to the population (DFO 2006), where allowable harm is defined as harm to the wildlife species that will not jeopardize its recovery or survival (DFO 2014). To permit otherwise prohibited activities under section 73 of SARA, or to exempt such activities in a Recovery Strategy, it must be demonstrated that the activity does not jeopardize the survival or recovery of the species.

Since the previous assessment (DFO 2006), new information on the White Shark population in the Northwest (NW) Atlantic has become available; suggesting a new assessment of allowable harm is warranted. The intent of this Science Response is to re-assess whether there is scope for allowable harm for White Shark in Atlantic Canadian waters. The specific question is: "*In accordance with the four criteria outlined in the Revised Framework for the Evaluation of Scope of Harm under Section 73 of SARA (DFO 2004), is there scope for harm (i.e., allowable harm) for White Shark in Atlantic Canadian waters?*". This information will inform the White Shark Recovery Strategy, the development and direction of the upcoming Action Plan, and the ongoing management of activities that may interact with this species.

This Science Response Report results from the Science Response Process of November 30, 2016, on the Evaluation of Allowable Harm for White Shark (*Carcharodon carcharias*) in Atlantic Canada.

Background

The four criteria outlined in SSR 2004/048 (DFO 2004) are:

1. The current population is neither so small that random factors threaten population viability nor so concentrated in space that that it is vulnerable to elimination by a catastrophic event.
2. The recent trajectory of the stock is stable or likely to be increasing, so that survival or recovery is not in jeopardy in the period when the permit is in place.
3. The known sources of human-induced mortality are unlikely to increase during the permitting period. This means that there is high confidence that the causes of human-induced mortality are under management control, monitored, and can be enforced effectively.
4. There is a relatively high likelihood that recovery goals will be achieved in biologically reasonable time frames with the activity present.

Canada

The recovery goal in the current version of the draft White Shark Recovery Strategy[1] (DFO, unpublished manuscript) is:

- Maintain or increase the population of White Sharks that frequents Atlantic Canadian waters.

This assessment must also comply with the 2015 Recovery Potential Assessment Guidelines[2] (DFO, unpublished manuscript). Element 22 in these guidelines relates to Allowable harm:

- Evaluate maximum human-induced mortality and habitat destruction that the species can sustain without jeopardizing its survival or recovery.

Guidance specifically states that information is intended to relate to whether harm to the species can be permitted under Section 73 of SARA and that allowable harm must be evaluated at the Designatable Unit (DU) assessed by COSEWIC (DFO, unpublished manuscript)[2]. For White Shark, this means the appropriate level at which to assess allowable harm is the population in the NW Atlantic Ocean, called the Atlantic population by COSEWIC (COSEWIC 2006). Options presented for allowable harm must not impact the survival or recovery of the DU. It is recognized that there may be activities that affect such a small proportion of the overall population that they do not amount to significant impacts at the population level.

The guidance given in the 2015 Recovery Potential Assessment Guidelines (DFO, unpublished manuscript)[2] on specific conditions which must be met in order to authorize activities under Section 73 of SARA are:

- Identify and quantify (to the extent possible), potential impacts on recovery goal(s) and recovery targets (e.g., whether allowable harm will result in a lower probability of and a longer time to recovery).
- Identify the likelihood of the activity to jeopardize the survival or recovery of the species.
- Report results in risk-based language, factoring in uncertainty.
- Provide options and recommendations regarding the potential permitting for allowable harm under section 73 of SARA, including rationales, relevant conditions and performance measures.

The habitat destruction component outlined in the 2015 Recovery Potential Assessment Guidelines (DFO, unpublished manuscript)[2] is not considered in this document, due to very limited information.

Analysis and Response

Criterion 1: Population is Neither Critically Small nor Spatially-Concentrated

Globally, White Sharks are distributed throughout the world's oceans, yet individuals seem to concentrate in temperate coastal areas (Fergusson et al. 2009). In Atlantic Canada, White Sharks are encountered very infrequently (DFO 2006; Appendix 1) suggesting that Canadian waters are the northern limit of the NW Atlantic population's range (COSEWIC 2006). However, even within Canadian waters, sightings and bycatch records encompass a large geographic area: from the coast off northern Newfoundland, along the edge of the continental shelf, into the

[1] DFO, unpublished manuscript. White Shark Recovery Strategy. Draft 2016.
[2] DFO, unpublished manuscript. Guidance for the completion of Recovery Potential Assessments (RPA) for Aquatic Species at Risk. 2015.

Laurentian Channel and as far south as Grand Manan and into the Bay of Fundy. Individuals are capable of swimming vast distances (Bruce et al. 2006) with documented transoceanic movements (> 10,000 km) as well as regular coastal movements in excess of 1000 km (Kohler et al. 1998, Hammerschlag et al. 2011). The population in the NW Atlantic is not spatially concentrated.

The White Shark population in the NW Atlantic Ocean is likely to be small; however, there are no current estimates of population size. Globally, the species is uncommon relative to other widely distributed sharks (Fergusson et al. 2009). Movement studies suggest that White Sharks are highly migratory and exhibit a diversity of behaviour patterns, yet show fidelity to specific aggregation sites on a seasonal or annual basis (Bruce et al. 2006, Chapple et al. 2011, Robbins et al. 2015, Andreotti et al. 2016). For several aggregation sites worldwide, photo-identification catalogues have been developed to enable abundance estimates from capture-recapture analyses. These estimates can be sensitive to the underlying assumptions or specific methodology used, leading to varying abundance estimates. For example, predictions of approximately 400 (Chapple et al. 2011) versus > 2000 (Burgess et al. 2014) White Sharks in the Northeastern Pacific Ocean, or approximately 400 individuals (Andreotti et al. 2016) versus approximately 900 individuals (Towner et al. 2013) in the Gansbaai region of South Africa.

In the NW Atlantic, increasing pinniped populations have led to an aggregation of White Shark off the coast of Cape Cod, Massachusetts. Research conducted from 2009 to 2013 demonstrates that some individuals exhibit site-fidelity over multiple years (Skomal and Chisolm 2014), similar to aggregations at which abundance has been estimated. It is possible that White Sharks traveling into Canadian waters originate from this aggregation, due to its geographic proximity. A photo-identification catalogue is currently being compiled for White Shark encountered off Cape Cod with the expectation that it will inform a population estimate. There are no known aggregation sites in Canadian waters for White Shark.

Criterion 2: Recent Trajectory is Stable or Likely to be Increasing

There is limited data available to estimate the population trajectory of White Shark, and the majority of it is fishery-dependent (Curtis et al. 2014). Using fishery-dependent data (such as catch-per-unit effort) as indicative of population abundance can be problematic given that many components of the fishing process affect measures of effort (Hilborn and Walters 1992). For White Shark, this has led to controversy in our understanding of population trajectories.

Baum et al. (2003) analyzed trends in the bycatch of multiple shark species recorded in the logbook data of the United States pelagic longline fleet targeting swordfish and tunas. They estimated White Shark to have declined by 79% over 14 years (95% Confidence Interval 59 – 89%), based on data collected during 1986 to 2000. The conclusions from this analysis were subsequently criticized on two main fronts for White Shark:

1. the selection of a single data set relative to multiple alternate sources, and
2. possible species misidentification affecting the prevalence of White Shark in the data from the Southern States and Caribbean (Burgess et al. 2005).

Baum et al. (2005) rebutted these major criticisms to conclude that their trend estimates were robust, and COSEWIC (2006) based their designation of status for White Shark largely on the Baum et al. (2003) trend estimates.

Recent genetic analyses from 35 individuals at 14 microsatellite markers suggest that the White Shark population in the NW Atlantic underwent a bottleneck in the mid- to late 20th century (O'Leary et al. 2014). This result is consistent with the population having undergone substantial

decline. Given the length of time necessary for genetic changes at neutral markers like microsatellites to manifest in a population (multiple generations; Landguth et al. 2010), this type of analysis would not be useful to infer more recent changes in abundance at present.

Specific to White Sharks, Curtis et al. (2014) evaluated trends in abundance from longline catch data (a fishery-independent survey and the observer program associated with the directed shark bottom longline fishery), two recreational fishing tournament time series, and historical sightings information) in the US. Collectively, data spanned 1800 to 2010. It is important to note that the data source used by Baum et al. 2003 was not updated or included. This analysis suggested a strong decline in abundance of White Shark throughout the 1970s and 1980s (63-73%), prior to the time period considered by Baum et al. (2003). More recently, it reveals a positive trend since the 1990s, which appears to be driven primarily by an increase in relative abundance of White Shark recorded by observers of the directed shark bottom longline fishery (1994-2010). The authors suggest that the increase coincides with the implementation of the first Fishery Management Plan for Atlantic sharks (1993) as well as the prohibition on landing White Shark during commercial fishing or in recreational tournaments (1997) in the US. In addition, it conforms well to the general increase in sightings since the early 2000s (Curtis et al. 2014). Curtis et al. (2014) do not suggest a magnitude of increase, merely that the trend is positive.

One criticism by Burgess et al. (2005) that Baum et al. (2005) did not adequately address was the change in management practices (*circa* 1993) affecting the pelagic logbook time series. Prior to 1993, participants in the directed shark fishery as well as in the directed tuna and swordfish fishery reported shark landings in the logbook for pelagic longlines, yet the directed shark fishery switched to using a new logbook during 1993-1994 (Burgess et al. 2005). Figure 2 panel B in Baum et al. (2003) shows a distinct difference in relative abundance of White Shark up to 1993 and beyond 1993. This discrepancy lends support to the idea that the decline in White Shark abundance was not as substantial as reported by Baum et al. (2003) and may have been an artifact of reporting practices. As argued by Burgess et al. (2005), participants in the longline fishery targeting sharks would be more likely to report all species of shark bycatch in comparison to participants in the tuna/swordfish fishery. The increasing trends reported by Curtis et al. (2014) would not be affected by this change in reporting. Also, the data sets included in the Curtis et al. (2014) analysis would not be as biased by misidentifications as well as potential under- and over-reporting of specific species as in the commercial logbooks, partially because these data were collected by trained personnel.

Criterion 3: Known Sources of Human-Induced Mortality are Unlikely to Increase

Targeted and bycatch fisheries are a well-understood source of human-induced mortality for White Sharks. Shark finning and the sale of other body parts as trophies is considered to be the greatest threat to White Sharks worldwide (Fergusson et al. 2009, DFO 2006). In response to this threat, White Shark has been afforded some of the highest levels of international protection for any elasmobranch species. White Shark has been listed as a prohibited species (i.e. no recreational or commercial harvest) in US waters since 1997 (Curtis et al. 2014). Cooperative international management is stipulated by their inclusion as a highly migratory species in Annex I of The United Nations Convention on Law of the Sea (UNCLOS). White Shark were listed on Appendix I and II of the Convention for the Conservation of Migratory Species (CMS) in 2002 and on Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) in 2005. Under both of these listings, signature countries agreed to strictly control trade in White Shark parts, with very few exemptions. Canada ratified listings under CMS in 2003, and listings under CITES in 2004 (DFO 2006). Collectively, these efforts are expected to have substantially reduced the global curio and fin trade of White Sharks. It is

highly unlikely that White Sharks from Canadian waters have ever contributed substantially to global markets, particularly since Canada banned shark finning in 1993 (DFO 2006). Such international management controls would prevent any development of a legitimate market for White Shark parts originating from Canada.

In Atlantic Canada, fishery impacts on White Sharks are thought to be limited to incidental captures leading to mortality. The recently updated list of historical encounters and bycatch records (Appendix 1) was used to identify fisheries with the potential for incidental captures. All records identified as captures by a fishery (rather than sightings or other interactions) were assumed to be mortalities unless the record specifically stated that the animal was released alive. Of the 31 animals that were likely mortalities, 17 of them were caught in coastal gill nets or weirs (Table 1). There were never more than 2 animals incidentally killed in a single year in this time series, occurring most recently in 1977 (Appendix 1); although it is possible that these 2 records refer to the same shark. There was a single record in the observer database from an incidental capture of a White Shark in a domestic fishery, occurring in a trawl set for cod in 1992 (Appendix 1). Although incomplete reporting and species misidentification would affect these data, incidental captures causing direct mortality to White Shark are very rare in Canadian waters.

Table 1. A summary of the incidental mortalities of White Shark from Canadian fisheries during 1874-2016.

Gear Type	Species Sought	Years	Records
Gill net	hake, mackerel, cod, not specified	1921-early 1990s	7
Weir	herring	1930-2011	10
Purse seine	mackerel	1934, 1950s	2
Rod and line	various	1938, 1953	2
Pelagic longline	tuna	1989	1
Trawl	cod, not specified	1956, 1971, 1992	3
Unknown	unknown	1938-1965, 2015	6

Potential interactions with Canadian fisheries are expected to occur primarily over the continental shelf, based on depth preferences reported from 564 observations of White Shark in US waters. The median reported depth of encounters was 30 m and 92% of observations occurred over the continental shelf in water < 100 m deep (Curtis et al. 2014). In terms of gear type, gill nets and weirs seem to pose the greatest potential for interaction with White Sharks in Atlantic Canada as well as for incidental mortality.

It is exceptionally difficult to evaluate the risk for incidental capture of White Shark in commercial marine fisheries. Gear types that have led to incidental mortality in the past are fished throughout waters on the Continental Shelf. The severity of concern for a given fishery would rely on evaluating its magnitude, spatial and temporal overlap with the distribution of White Shark, and likelihood of capture or incidental mortality. A similar type of undertaking was recently attempted for inner Bay of Fundy (iBoF) Atlantic Salmon (DFO 2016) because their seasonal distribution within the iBoF and adjacent watersheds is known. However, it is informative that even within this spatially-restricted area, approximately 100 federally licensed fisheries were identified that had to be scored in a risk-based framework. DFO still concluded that data on bycatch within the iBoF is lacking (DFO 2016). There would be many more federally-licensed fisheries that would have to be evaluated relative to White Shark as compared to iBoF Atlantic Salmon. The magnitude of effort that would be required for this type

of an evaluation is not justified by the low number (31) of animals that have been incidentally killed in the last 100+ years.

Relative to other shark species in the Maritimes Region, the potential for incidental capture of White Shark by commercial marine fisheries is very low. One of the most comprehensive reviews of discards in Canadian commercial fisheries considered Northwest Atlantic Fisheries Organization Divisions 4VWX and 5YZ (Maritimes Region) during 2002 to 2006 (Gavaris et al. 2010). White Shark is not mentioned in Gavaris et al. (2010) due to the lack of captures during at-sea monitoring and the associated lack of records in the Industry Surveys Database (ISDB) for those years. However, this is not true for other shark species. Similar to White Shark, the populations of Shortfin Mako (*Isurus oxyrinchus*), Blue Shark (*Prionace glauca*), and Porbeagle Shark (*Lamna nasus*) are widely distributed throughout the NW Atlantic and have the potential to interact with a wide variety of commercial marine fisheries. Bycatch of these three species was regularly observed during this time period and attributed primarily to the swordfish longline fishery in 4VW and 4X5Y (Gavaris et al. 2010). It is important to note that the years considered in this assessment pre-date the closure of the directed porbeagle fishery and that substantial management changes aimed at reducing bycatch by the longline fleet have been implemented since 2006.

Considering the sporadic nature of incidental mortalities in the historical encounters and bycatch data, captures of White Shark can be considered a chance event. Since the 1960s there have been a maximum of 3 incidental mortalities recorded per decade (Appendix 1). Commercial fishing activity throughout Atlantic Canada was much higher and subject to less stringent management controls in previous decades than it is today, largely because of current moratoriums on specific species at low abundance (e.g. Atlantic cod; Gavaris et al. 2010). Also, White Sharks are thought to have been much more numerous in the 1950s and 1960s than today (Curtis et al. 2014). If the rate of incidental capture of White Shark in Canadian commercial fisheries was proportional to the amount of fishing activity, mortalities should have been more frequent in previous decades than today. Furthermore, the lack of relationship suggests that increased capacity in current fleets would not lead to measurable increases in the rate of incidental captures of White Shark.

In relation to other threats, COSEWIC (2006) identified that bioaccumulation of pollutants may adversely affect populations of White Shark, including the one in the NW Atlantic (COSEWIC 2006, Mull et al. 2013). Shark species accumulate toxins readily due to their high trophic position, life history characteristics (slow growth and longevity), and large, lipid-rich livers (Schlenk et al. 2005). There is also evidence that female White Shark transfer contaminant loads to their offspring through oophagy, when embryos feed on sequentially-ovulated unfertilized eggs (Mull et al. 2013). However, changes in survival rates or other life history parameters related to organochloride contaminant loads have not been demonstrated to date (COSEWIC 2006, Mull et al. 2013).

Criterion 4: Likelihood of Achieving Recovery Goals with Harm Permitted

The recovery goal incorporated into the draft White Shark Recovery Strategy[3] (DFO, unpublished manuscript) is focused on maintaining or increasing the number of White Sharks that frequent Atlantic Canadian waters. This could be accomplished by increasing the proportion of the total NW Atlantic population that travels northward (i.e. animals shift their distribution patterns, becoming more commonly found in Canadian waters) and/or total abundance in the NW Atlantic increases, proportionately increasing the number of animals frequenting Canadian

[3] DFO, unpublished manuscript. White Shark Recovery Strategy. Draft 2016.

waters. Altering White Shark distribution patterns in isolation of changes in abundance is not consistent with the intent of the *SARA*. Therefore, analyses relative to this criterion focuses on how human-induced mortality in Canadian waters may influence total abundance of White Shark in the NW Atlantic. In this context, activities that would have a high probability of leading to population decline would jeopardize the survival and recovery of White Shark.

The allowable harm scenarios assessed in this report relate specifically to fisheries-induced mortality from bycatch in any Canadian commercial fishery. No attempt was made to partition harm among individual fisheries. Because relatively little is known about the life history of White Sharks (Curtis et al. 2014) particularly in Canadian waters (DFO 2006), alternate scenarios are presented based on somewhat contradictory evaluations of White Shark life history.

Sustainable Mortality Rate

In the absence of an abundance time series or an estimate of population size, allowable harm can be evaluated in the context of changes to a population's trajectory (i.e. changes in its ability to grow in size). The capacity for population growth (r) for a given species can be estimated using life table analyses. As data inputs, life table analyses use age-structured estimates of survival rates, information on the timing of maturation, and fecundity. It is well suited for use in shark species given their well-defined reproductive cycle and high rates of survival (Cortés 1998). From life tables, it is also possible to determine the critical level of human-induced mortality (F_{crit}) at which population growth is zero. At this level of mortality, the population should theoretically be maintained at a constant abundance, but would have no capacity to increase in size. As such, F_{crit} could be considered the maximum level of mortality the population could sustain. Above this fishing mortality rate, the population would be driven to extinction (Campana et al. 2008).

There are multiple methods for estimating the capacity for population growth from life table data (Cortés 2016). In this Science Response, a derivation of the Euler-Lotka equation that is based on exponential population growth has been used. This is a density-independent model, where the estimate of r is often taken to represent the intrinsic rate of population increase ($r_{intrinsic}$ or r_{max}), which would be the maximum rate the population can increase from a severely depleted population size (Gedamke et al. 2007, Zhou et al. 2012). For White Shark, it is unknown if the current population in the NW Atlantic can be considered severely depleted, thus the estimates of r will be slightly smaller than r_{max}. Relative to the potential for human-induced mortality, this underestimate would make the analyses precautionary. Also, in the simulations presented in the following section, any underestimate of r will be small relative to the range of values that have been profiled over.

The Euler-Lotka equation cannot be solved analytically, so r is approximated by minimizing:

$$1 = \sum_{x=0}^{A} e^{-rx} m_x l_x$$

Here, A is the maximum age, l_x is survival to age x ($l_0 = 1$), and m_x is the expected reproductive output at age x. Estimating survival to each age relies on estimating the natural mortality rate (M) of the population, which was assumed to be constant in these analyses (Campana et al. 2008). This is a female-only model relative to reproductive output, which is calculated by age from female age-at-maturity, the sex ratio (here assumed to be 50:50), and fecundity of females.

Age validation is difficult for shark species, and this has led to distinctly differing viewpoints on how White Sharks grow, the age/size at which they mature, as well as their overall longevity.

Original age estimates suggest a short lifespan and early maturity in White Shark, with females in the Northeast Pacific beginning to mature at Age 7 and with a life span of approximately 35 years (Tanaka et al. 2011, Andrews and Kerr 2015). In contrast, recent research in the NW Atlantic based on bomb radiocarbon suggests that White Sharks grow slowly and have a long lifespan, with females maturing into their 30s (Natanson and Skomal 2015) and maximum ages being 70+ years (Hamady et al. 2014, Natanson and Skomal 2015). In this report, an evaluation of allowable harm is presented relative to both estimates of longevity (parameters in Table 2), although it is likely that the longer lifespan and later maturity is more representative of the population in the NW Atlantic.

Table 2. Life history parameter values used in the life table analyses.

Lifespan	Age-at-Maturity	Maximum Age	Reproductive Rate (years)	Average Litter Size (pups)	Instantaneous Natural Mortality
Short	10	40	2	8	0.112
Long	30	70	2	8	0.063

The natural mortality rate was approximated using the geometic mean regression equation for mammals done by Hoenig (1983). This regression technique is better suited to data that contain uncertainty in the estimate of maximum age than traditional estimation based on least squares (Kenchington 2014). The relationship for mammals is expected to better approximate an elasmobranch life history than the equivalent type of relationship that has been estimated for fishes.

Assuming a short lifespan, the rate of population increase (r) for an unfished population is 0.101, the estimate of the number of female offspring produced by a single adult female during their lifetime is 5.6 and the annual reproductive rate in the population is 0.367. The annual reproductive rate is defined as the expected number of reproducing animals produced by each reproducing animal per year, after a lag of a years, where a is the age-at-maturity (Myers et al. 1999). The annual reproductive rate is low as compared to most fishes (Myers et al. 1999), yet the estimate of r is higher than many large-bodied shark species (Smith et al. 1998, Cortés 2016). Calculating generation time (G) as a function of expected reproductive output and the population growth rate, $G = log(2 * 5.6)/0.101$, gives an estimate of 24 years.

Assuming a long lifespan, the rate of population increase (r) is lower, at 0.035. However, the expected number of females produced per adult female is only slightly less (4.4), given the longer number of years over which an animal is mature (40) as compared to the shorter lifespan (30). The annual reproductive rate for the population is 0.220 and the generation time estimate (calculated as above) is 62 years.

Human-induced mortality was incorporated into the calculation of age-specific survival to determine the maximum level the population could sustain, above which it would be driven to extinction (F_{cmt}). This value is estimated by finding the value of F such that the net reproductive rate equals one:

$$\sum_{x=0}^{A} m_x l_x = 1$$

Survival by age becomes:

$$l_x = \prod_{i=0}^{x-1} e^{-(M_i+F_i)}$$

For this calculation, it is necessary to define the component of the population that is vulnerable to fishing pressure. In this report, fishery selectivity was assumed to be knife-edged at Age 2, similar to the assumption for basking shark in Campana et al. (2008). Looking at the mortalities in the historical sightings and bycatch records, the majority appear to be juveniles on the basis of size, although larger individuals (> 5 m; presumably adults) have been captured in weirs (Appendix 1). Based on the two sets of life history parameters and assuming that fishing selectivity is knife-edged at Age 2, F_{crit} is estimated to be 0.116 and 0.037 for the short and long lifespans, respectively.

To put this value in context, the size of the population (N_{crit}) that would be required for F to equal F_{crit}, given average removals, was calculated:

$$N_{crit} = \left(\frac{removals}{1 - e^{-F_{crit}}}\right)\left(\frac{\sum_{x=0}^{A} l_x}{\sum_{x=sel}^{A} l_x}\right)$$

The second term on the right of the equation corrects for the proportion of the population that is subject to fishing mortality. It is appropriate when selectivity is knife-edged and mortality due to bycatch is 100% (Campana et al. 2008). The mean annual number of discards of White Sharks in Canada is zero, considering the years 1950 to 2016. However, 1 animal was assumed for these calculations. For the short and long lifespans, N_{crit} estimates were exceptionally low (13 and 33 animals, respectively). What this means is that mortality from bycatch in Canadian waters is so low relative to the potential for population growth of White Shark that extremely small population sizes in the NW Atlantic can support this level of removals without going deterministically extinct.

Population Response to Allowable Harm

The DFO guidelines for Recovery Potential Assessments[4] (DFO, unpublished manuscript) require an evaluation of uncertainty when assessing allowable harm. For White Shark, the demographic parameters used in the previous analysis are not known with precision, given that they are based on very limited sampling (e.g. Hamady et al. 2014). Similarly, an argument can be made that mortalities due to fishing activities are uncertain and have the potential to be under-reported. Therefore, a demographic analysis using Monte Carlo methods was done to calculate uncertainty in the potential population growth rate (r) as well as in population trajectories subject to varying levels of human-induced mortality. Similar analyses have been done for North Atlantic harbor porpoise (Caswell et al. 1998), dolphins off Patagonia, Argentina (Dans et al. 2003), and basking shark in Atlantic Canada (Campana et al. 2008). This analysis was arbitrarily restricted to the years 1950-2016, to represent a time period largely characterized by industrialized commercial marine fisheries in Canada.

The population of White Shark in the NW Atlantic is considered to be at low abundance (COSEWIC 2006) so density-dependence was ignored in this analysis. Assuming exponential population growth, the population size in a given year (N_t) equals the population size in the previous year multiplied by r, minus the number of human-induced mortalities in the previous year (D_{t-1}):

[4]DFO, unpublished manuscript. Guidance for the completion of Recovery Potential Assessments (RPA) for Aquatic Species at Risk. 2015.

$$N_t = e^r N_{t-1} - D_{t-1}$$

If a time series of the number of human-induced mortalities is available and *r* is known, an abundance time series can be calculated by projecting backwards in time from a starting population size:

$$N_{t-1} = \frac{N_t - D_{t-1}}{e^r}$$

This formulation assumes that birth and natural deaths occur first and that incidental deaths occur afterwards in each year. Although this assumption would exaggerate the effect of incidental mortalities relative to modeling these as a continuous process throughout the year, this is not considered an issue given the extremely low numbers of incidental mortalities.

As previously discussed, life history parameters are quite uncertain for White Shark and there is no estimate of current abundance. Therefore, Monte Carlo sampling from assumed distributions for the input parameters was used to generate a range of estimates of *r* as well as a range of estimates of current population size. The Monty Carlo sampling works by assuming bounds (minimum and maximum values) for specific life history parameters and then randomly sampling within these bounds, based on an assumed distribution. For these simulations, random draws of integer values came from uniform distributions for population size in 2016, age-at-maturity for females, female litter size, gestation period, maximum age, and natural morality (Table 3). To make variability affecting the 2 longevity scenarios comparable, the minimum and maximum bounds for age-at-maturity, natural mortality and maximum age were set relative to a similar percentage above and below the mean estimate, rounded to an integer value. For example, the range of 8-11 is 16% above and below a mean estimate of 9.5 for age-at-maturity (Cortés 2016) for the short life history; the closest integer value that is approximately 16% of a mean estimate of 30 for age-at-maturity (Natanson and Skomal 2015) would be 5, giving a range of 25-35 for the long life history.

Table 3. Assumed bounds for the Monte Carlo sampling of life history parameters to estimate r, as well as assumed bounds for population size in 2016 for White Shark in the Northwest Atlantic.

Lifespan	Parameter	Minimum	Maximum
Short	Population size	100	1000
Short	Age-at-maturity	8	11
Short	Female litter size	2	6
Short	Gestation period	2	3
Short	Maximum age	35	45
Short	Natural mortality	0.062	0.162
Long	Population size	100	1000
Long	Age-at-maturity	25	35
Long	Female litter size	2	6
Long	Gestation period	2	3
Long	Maximum age	60	80
Long	Natural mortality	0.053	0.073

For the 2 lifespan scenarios, sets of randomly-drawn parameter values were used to calculate *r* using the Euler-Lotka method described above. Some of the resulting values for *r* were less than 0 and some were unrealistically high, so a limit of 2 times the deterministic estimate of *r* was used as a cutoff value, equating to 0.2 and 0.07 for the short and long lifespan scenarios,

respectively. Simulations that produced values of *r* that were above this cutoff or below zero were discarded.

From each simulated value for *r*, an abundance time series for the years 1950-2016 was calculated, based on a randomly drawn population size for 2016 (Figure 1) and incorporating human-induced mortalities from 1950-2016. Remember that the abundance trajectory is projected backwards. Because an exponential model was assumed to represent the potential for population growth, an exponential decay model (i.e. a lognormal regression) can be fit to determine if the abundance time series was increasing or decreasing:

$$ln(N_t) = \alpha + \beta t$$

Here, β represents the instantaneous rate of change in population size, where positive values indicate an increasing population.

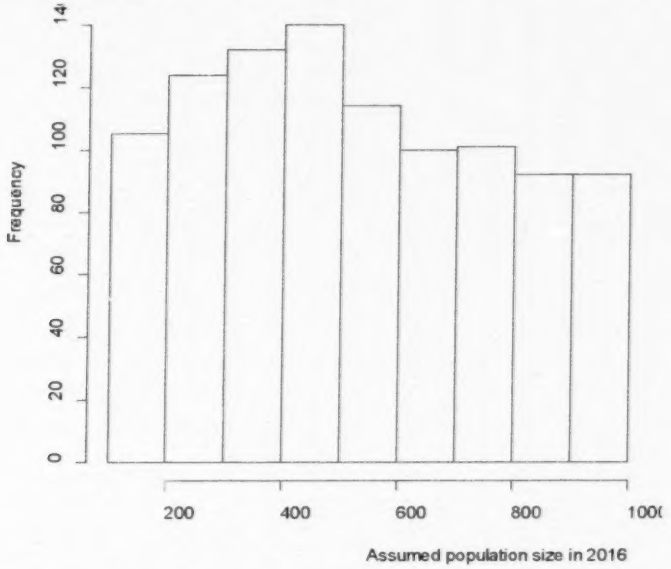

Figure 1. Histogram for the distribution of population sizes for 2016 (N=1000) considered in the simulations, representing random draws from a uniform distribution for sizes 100-500 (N=500); and 500-1000 (N=500). Random sampling was done this way to avoid over-representing large population sizes in the simulations, reflecting the assumption that White Shark populations are at low abundance.

To encompass variability that may arise due to misidentifications of White Shark or under-reporting of human-induced mortality, 6 different mortality scenarios were considered in the backwards projections: no fisheries removals, the historical bycatch records from 1950-2016 (21 total removals; Appendix 1), one removal per year from 1950-2016 (67 removals), 3 removals per year from 1950-2016 (201 removals), and 10 removals per year from 1950-2016 (670 removals) and 20 removals per year from 1950-2016 (1340 removals). Relative to the historical record of human-induced mortalities for White Shark in Canada, this last scenario represents an increase of > 5400% (Table 4). Although this level of removals from Canadian fisheries is unrealistically high, it was included in order to be able to evaluate N_{crit} for the NW Atlantic population as a whole.

Table 4. Comparison of the 6 mortality scenarios considered in terms of the number of human-induced mortalities per year, the total number of human-induced mortalities over the time series (1950-2016), and the percent difference of each scenario relative to the actual record of removals.

Mortalities (Number/Year)	Total	Difference (%)
None	0	-100
Actual	21	0
1	67	179
3	201	738
10	670	2692
20	1340	5483

Incorporating life history variation in the Euler-Lotka calculation of r for the short lifespan scenario gave a median value of $r = 0.08$ (10^{th}, 90^{th} quantiles = 0.029, 0.148). This is slightly lower than the deterministic calculation of 0.101. The median level of fishing mortality above which the population would be driven to extinction (F_{crit}) was 0.092 (10^{th}, 90^{th} quantiles = 0.033, 0.173), as compared to the deterministic value of 0.116. Accounting for variability in life history parameters lowers the expectation of the population's potential for growth, as well as the level of human-induced mortality that it can sustain.

The backwards projections of population growth under varying levels of human-induced mortality suggest that mortality has to be very high if the White Shark population in the NW Atlantic has been declining from 1950 to 2016. Given the expected capacity for population growth, nearly all simulations suggest that population sizes were smaller in 1950 than in 2016 (left panels, Figure 2) and that the population trajectory over the years 1950 to 2016 was positive (right panels, Figure 2). This means that even theoretical removals of 20 animals per year from Canadian fisheries are not high enough relative to the potential for population growth for White Shark in the NW Atlantic to reliably lead to population decline, assuming an exponential growth model. Removals from Canadian fisheries have negligible influence on the population trajectory of White Shark in the NW Atlantic, even if they are increased to unrealistically high levels. Assuming that the White Shark population in the NW Atlantic has undergone substantial declines (COSEWIC 2006), human-induced mortality from Canadian fisheries must affect a very small component of the total population, such that their impact is negligible.

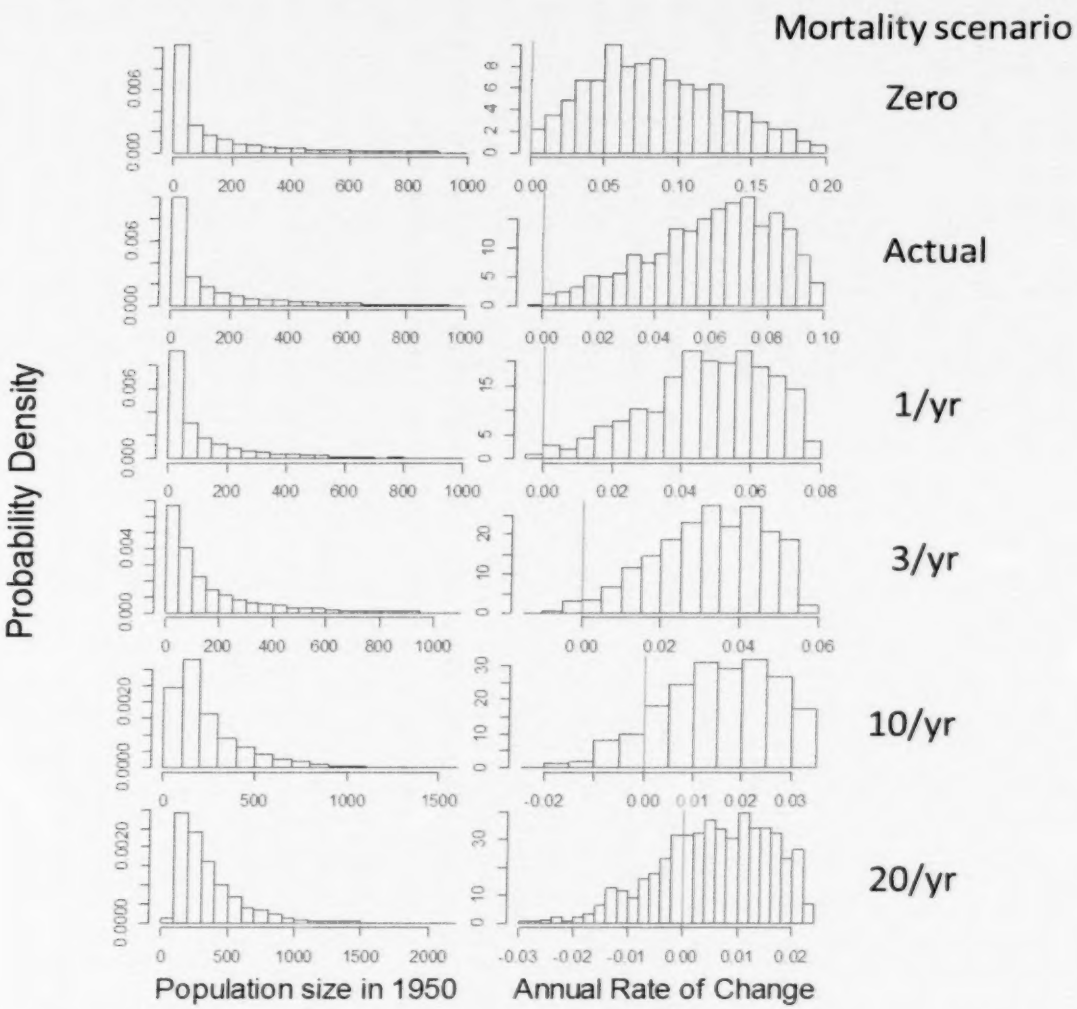

Figure 2. Predicted population size in 1950 and the annual rate of change in population size for White Shark assuming a short lifespan and an exponential model for population growth for the six mortality scenarios listed in Table 4. Positive and negative annual rates of change are separated by the vertical red line.

Incorporating life history variation for the long lifespan scenario gave a median value of $r = 0.028$ (10^{th}, 90^{th} quantiles = 0.011, 0.042). This is slightly lower than the deterministic calculation of 0.035. The median level of fishing mortality above which the population would be driven to extinction (F_{crit}) was 0.029 (10^{th}, 90^{th} quantiles = 0.011, 0.045), as compared to the deterministic value of 0.037. Again, accounting for variability in life history parameters lowers the expectation of the population's potential for growth as well as the level of human-induced mortality that it can sustain.

Even if the productivity of the White Shark population in the NW Atlantic is much lower (as in the long lifespan scenario), removals from Canadian fisheries need to be substantial on an annual

basis before simulated population sizes in 1950 must be larger than those in 2016 (left panels, Figure 3) and population trajectories tend to be negative (right panels, Figure 3). In this situation, the impact of actual removals from Canadian fisheries is still negligible, but the population in the NW Atlantic is much more susceptible in general to increasing levels of bycatch mortality.

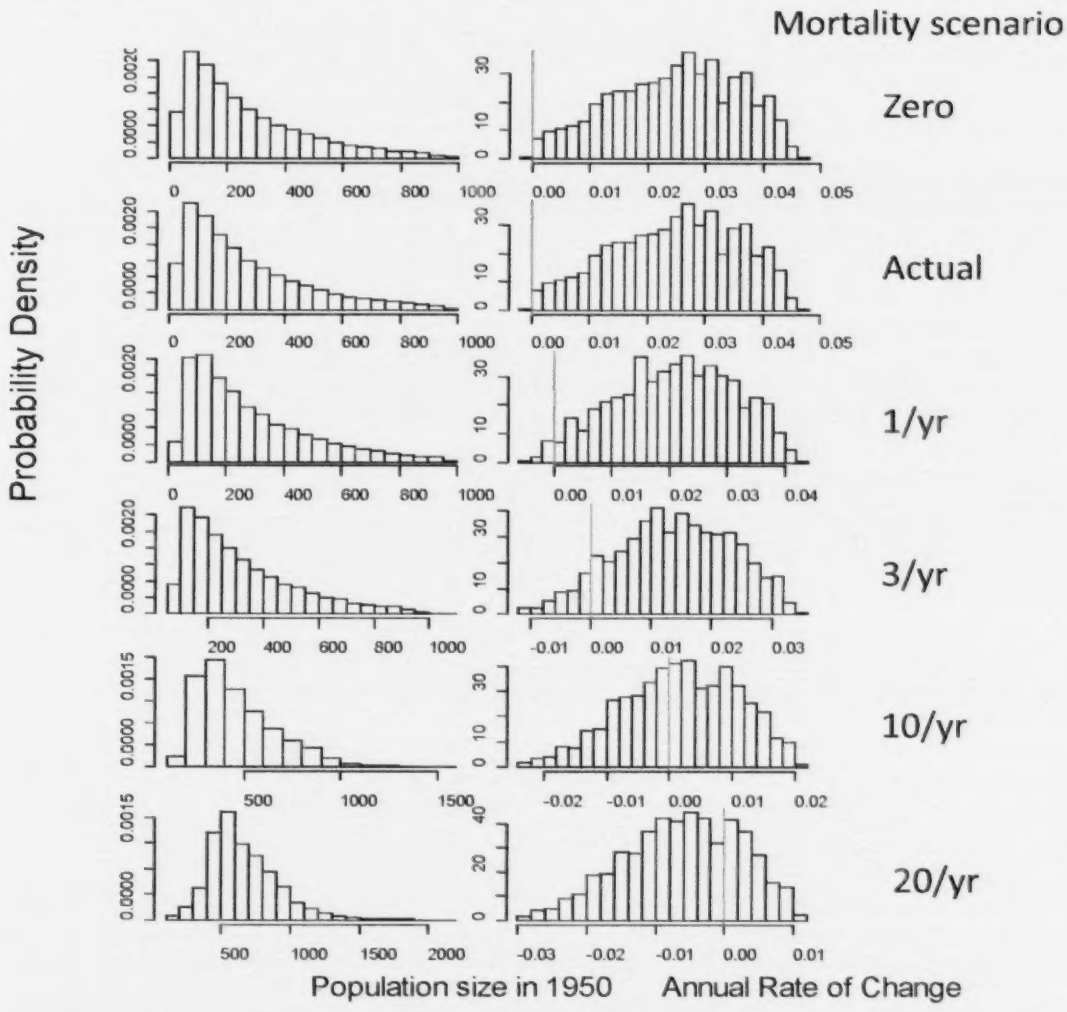

Figure 3. Predicted population size in 1950 and the annual rate of change in population size for White Shark assuming a long lifespan and an exponential model for population growth for the six mortality scenarios. Positive and negative annual rates of change are separated by the vertical red line.

Feasibility of Population Recovery with Allowable Harm

To assess the feasibility of future population recovery under each mortality scenario, the mean annual exploitation rate (u) can be calculated from the backwards projections and incorporated into Euler-Lotka re-calculations of r within each simulation ($r^{forward}$). The population can then be projected into the future (3 generations) from the randomly-drawn population size for 2016. In

other words, the rate of population increase in the absence of fishing is reduced to account for historical fishing mortality and then is used to predict the population trajectory under that level of allowable harm. As such, these simulations evaluate the likelihood of population recovery under various levels of allowable harm.

For greater biological realism in the projections, autocorrelated annual variability was incorporated into $r^{forward}$ following the general approach of Hilborn (2001). The deviates (w) were calculated as:

$$w_t = w_{t-1}d + w_t^*\sigma$$

where

$$w_t^* \sim N(0,1)$$

Deviates were added to the original Euler-Lotka estimate of $r^{forward}$ to produce a time-varying vector ($r_t^{forward}$). Annual variability (σ) and environmental autocorrelation (d) were given assumed values of 0.07 and 0.03 in the forward projections for the short and long lifespan scenarios, respectively. These values result in slightly increased variability in the distribution of $r^{forward}$ versus r.

Population trajectories were projected forward for 72 years and 186 years for the short and long lifespan scenarios, respectively (3 generations) and were summarized relative to the proportion of the trajectories which were declining. These predictions are sensitive to the level of variation and autocorrelation assumed for each life history (i.e. 0.07 vs 0.03 for the short and long lifespans, respectively). Thus, results from the two lifespan scenarios are not directly comparable. Estimates of the likelihood of population decline are also sensitive to the distribution of assumed population sizes in 2016 (Table 3, Figure 1). If abundance of White Shark in the NW Atlantic is less than the distribution considered (i.e. < 100 animals), the probability of population decline will be underestimated from these simulations. Conversely, if abundance in the NW Atlantic is greater than the distribution considered (> 1000 animals), the probability of population decline will be overestimated.

Accounting for the historical record of human-induced mortalities, approximately 28% of the simulated population trajectories were predicted to decline over 3 generations for the short lifespan scenario (Table 5). Increasing the total number of White Shark incidental mortalities to 3x the historical record (i.e. one mortality annually over 67 years) had very little influence on predictions, increasing by 3% (to 0.291). Human-induced mortalities needed to be high on an annual basis (i.e. greater than 10 animals per year) in order for more than 50% of the predicted population trajectories to be in decline (Table 5). Even when removals became 20 per year, the number of simulations that were predicted to be in decline over three generations did not even double. If the White Shark population in the NW Atlantic is characterized by relatively early maturity and a shorter lifespan, it is predicted to be fairly productive in terms of its ability to grow in size. This characteristic is reflected in the distribution of $r^{forward}$ values as well as in the relatively small median sizes for N_{crit} in any mortality scenario (Table 5).

Table 5. Summary of the population projections over three generations for the short and long lifespan scenarios relative to life history variability in future population growth rates ($r^{forward}$), critical population size (N_{crit}) and the proportion of the trajectories that are declining.

Lifespan	Removals (#/yr)	$r^{forward}$ 10th	$r^{forward}$ Median	$r^{forward}$ 90th	N_{crit} 10th	N_{crit} Median	N_{crit} 90th	Prop. Declining
Short	Actual	0.019	0.103	0.171	9	16	41	0.283
Short	1	0.017	0.102	0.169	9	16	41	0.291
Short	3	0.013	0.097	0.165	28	49	124	0.392
Short	10	0.003	0.087	0.154	95	163	414	0.502
Short	20	-0.006	0.078	0.146	190	326	829	0.594
Long	Actual	0.002	0.035	0.066	28	42	104	0.037
Long	1	0.000	0.034	0.065	28	42	104	0.076
Long	3	-0.003	0.031	0.061	83	125	311	0.211
Long	10	-0.012	0.022	0.052	278	415	1038	0.525
Long	20	-0.020	0.014	0.044	557	831	2076	0.723

Results from the long lifespan scenario predict greater sensitivity to changes in the level of human-induced mortality. Although only 4% of population trajectories were predicted to decline assuming the historical record of bycatch mortality, this percentage doubles assuming one mortality per year, increases by >5.5 times assuming three mortalities per year, increases by > 14 times assuming 10 mortalities per year and increases by > 19 times assuming 20 mortalities per year (Table 5). If removals of White Shark are 20 animals per year, the median estimate for N_{crit} is fairly large, at 831 animals, and 72% of population trajectories are predicted to decline over three generations (Table 5).

Conclusions

In relation to the four criteria outlined in DFO (2004), there appears to be some scope for allowable harm for White Shark in Canadian waters.

The spatial distribution and movement patterns of White Sharks make them relatively invulnerable to localized catastrophic events, even though total population size is expected to be relatively small.

Although there is substantial uncertainty about the recent population trajectory for White Shark in the NW Atlantic, the best available and most current trend estimate suggests that the population has been increasing since the 1990s. However, the magnitude of increase is uncertain. Indirectly, the development of a fishery-independent monitoring program for White Shark off Cape Cod in recent years (since 2009) supports the idea that White Shark are becoming more abundant in waters adjacent to Canada and may be more abundant in Canadian waters.

The main cause of human-induced mortality in Canadian waters was identified as incidental captures by marine commercial fisheries. In terms of gear types, weirs and gillnets appear to have the highest potential for interaction. However, there was no apparent relationship between the extent of commercial fishing and incidental mortality of White Shark (1950-2016). If incidental mortality in Canadian waters represents a chance event, it is unlikely to increase markedly in relation to future changes in commercial fishing activities.

There is a high likelihood that recovery goals can be achieved under various scenarios of allowable harm for White Shark in Canada, even if annual bycatch mortality increases from historical estimates. However, this conclusion directly depends on incidental mortality remaining low elsewhere in the population's range, as suggested by the positive population trajectory in US waters. Considering data from 1950-2016, mortality from bycatch in Canadian waters was so low relative to the potential for population growth that extremely small population sizes of White Shark in the NW Atlantic can support this level of removals without going deterministically extinct. This conclusion holds under markedly different evaluations of White Shark longevity. Given the low predicted sizes for N_{crit}, it is probable that Canadian mortalities have affected a small proportion of the overall population in the NW Atlantic. Accounting for life history variation and projecting the population into the future (3 generations) suggests that mortality would have to increase substantially before the majority of population trajectories (i.e. > 50%) would be predicted to decline.

The potential for allowable harm to White Shark should be re-evaluated if new information becomes available. A population size estimate in the NW Atlantic or new life history information could reduce uncertainty in the simulations of population recovery under varying levels of harm. Furthermore, allowable harm should be re-evaluated if future reporting in SARA logbooks demonstrates a persistently higher level of incidental mortality than that which occurred during 1950-2016. For example, if > 3 incidental mortalities of White Shark were reported each year for 5 years. Lastly, allowable harm from Canadian commercial fisheries should be re-evaluated if there is any indication of a change in the population's trajectory. If activities occurring outside of Canadian waters have substantially reduced the population's ability to grow in size, even low levels of allowable harm may negatively affect populations. The conclusions in this document are valid when the capture of White Shark represents a rare chance event and when activities occurring outside of Canadian waters are not causing population decline.

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Appendix 1

Historical record of White Shark sightings and bycatch in Atlantic Canada compiled and provided by Aimee Gromack (September 2016). (Notes: * = not authenticated; + = possibly the same shark; - = Unknown / data not collected; n/a = not applicable). The last column (Incidental mortality) identifies which animals were assumed to be mortalities in this assessment, as summarized in Table 1.

	Location	Date	Length (m)	Weight (kg)	Sex	Alive/ Dead	Gear Type	Comments	Reference	Inci. Mor.
1	St. Pierre Bank, NS	1873 or 1874	3.9	-	-	-	-	Teeth in attacked dory.	Putnam (1874)	No
2	Off Hubbard Cove, St. Margaret's Bay, NS	June 27, 1920	4.6	-	-	-	-	Tooth scrapes on attacked dory. Reported by fish harvester.	Piers (1934)	No
3	Georgetown PEI	September 17, 1921	2.1	272	-	dead	mackerel net	Caught by Capt Sam Hemphill off Georgetown PEI.	The Guardian (1921)	Yes
4	Georgetown PEI	September 17, 1921	2.7	453	-	dead	mackerel net	Caught by Capt Sam Hemphill off Georgetown PEI.	The Guardian (1921)	Yes
5*	White Head Island, near Grand Manan, NB	June (mid), 1930	11.3	-	-	dead (killed)	herring weir	Size suspect of basking shark, though teeth reported taken. Length likely over-estimated.	Vladykov and& McKenzie (1935)	Yes
6	16 km NW of Digby Gut, NS	July 2, 1932	4.6	-	-	alive	-	16 km NW, attacked fisherman and son in their 7.6 m boat, confirmed by teeth.	Piers (1933)	No
7*	Harbour de Loutre, Campobello Island, NB	November 22, 1932	7.9	-	-	-	herring weir	Trapped in herring weir.	Piers (1933)	Yes
8	French Village, NS	August 11, 1934	4.57	680.4	M	-	mackerel purse seine	Reported in news article by fish harvester. Mature.	Joyce pers. comm. (2016)	Yes

	Location	Date	Length (m)	Weight (kg)	Sex	Alive/ Dead	Gear Type	Comments	Reference	Inci. Mor.
9	Wedgeport, NS	August, 1938	2.6	196	M	-	rod and line	Caught on rod and line by Ms. Micchael Lerner.	Anon (1940) in Templeman (1963)	Yes
10	Whale Head, N shore, St. Lawrence River	August, 1938	-	-	-	-	unknown	n/a	Vladykov and McAllister (1961)	Yes
11	Isle Caribou, N shore, St. Lawrence River	August, 1942	2.7	-	-	-	unknown	n/a	Vladykov and McAllister (1961)	Yes
12	Isle Caribou, N shore, St. Lawrence River	August, 1943	3	-	-	-	unknown	n/a	Vladykov and McAllister (1961)	Yes
13	Deer Island, NB	August 24, 1949	3.87	590	F	-	herring weir	Trapped in herring weir; immature.	Scattergood et al. (1951)	Yes
14	Portneuf River estuary, N shore, St. Lawrence River	August 27, 1949	4.6	-	-	-	unknown	Shot by W.B. Scott.	Templeman (1963)	Yes
15	French Village, NS	1950s	-	-	F	dead	mackerel purse seine	Juvenile. Reported by fish harvester. Caught in trap.	Joyce pers. comm. (2016)	Yes
16	Between Passamaquoddy Bay and Grand Manan, NB	August 20, 1952	4.3	-	-	-	-	Observed attack on porpoise.	Day and Fisher (1954)	No
17	Off Fourchu, Cape Breton Island, NS	July 9, 1953	3.7	-	-	alive	-	Teeth in attacked dory; dory attacked and sunk.	Day and Fisher (1954)	No
18	Wedgeport, NS	July 9-10, 1953	2.4	-	M	-	rod and line	Caught by tuna fisherman.	Day and Fisher (1954)	Yes
19	La Have Islands, NS	August 12, 1953	4.7	-	-	-	herring trap (weir)	Caught in herring trap.	Day and Fisher (1954)	Yes

	Location	Date	Length (m)	Weight (kg)	Sex	Alive/ Dead	Gear Type	Comments	Reference	Inci. Mor.
20	St. Croix River, near Dochet Island between ME and NB	August 25, 1953	-	-	-	-	-	Observed attack on seal.	Day and Fisher (1954)	No
21	Mace's Bay, Bay of Fundy, NB	August 3, 1954	2.6	-	-	-	herring weir	Trapped in herring weir.	Leim and Day (1959)	Yes
22	Maces Bay, NB	September 10, 1954	4.87	-	F	-	herring weir	Caught in herring weir.	Hogans and Dadswell (1985)	Yes
23	Ireland Bight, Hare Bay; depth 26 m	August 10, 1956	3.7	-	-	-	cod trap	Teeth in codtrap leader.	Templeman (1963)	No
24	SE Grand Bank (44°30'N, 50°12'W)	August, 1956	3.7-4.6	-	-	-	otter trawl	Spanish otter trawl.	Templeman (1963)	Yes
25	Northumberland Strait, 13 km off Wallace, NS	July 30, 1962	3	-	-	-	hake gillnet	Tooth examined by L.R. Day.	Templeman (1963)	Yes
26	Northumberland Strait, 13 km off Wallace, NS	August (1st week), 1962	2.7	-	-	-	hake gillnet	ID by W.G. Smith, fishery officer.	Templeman (1963)	Yes
27	Wallace, NS	August, 1962*	6	-	-	-	hake gillnet	Escaped from gillnet.	Templeman (1963)	No
28	Wallace, NS	September, 1962*	6	-	-	-	hake gillnet	Escaped from gillnet	Templeman (1963)	No
29	Noel, Minas Basin, NS	Sept. 2, 1965	-	-	-	dead	unknown	Reported by fish harvester. Caught in net or handline.	Warren Joyce, pers. comm. (2016)	Yes
30	Passamaquoddy Bay, between ME and NB	1969	-	-	-	-	-	Observed attack on porpoise.	Arnold (1972)	No

	Location	Date	Length (m)	Weight (kg)	Sex	Alive/ Dead	Gear Type	Comments	Reference	Inci. Mor.
31	Passamaquoddy Bay off Leonardville, Deer Island, NB	August 13-14, 1971	4.3	-	F	-	otter trawl	Caught in otter trawl.	Scott and Scott (1988)	Yes
32	Letite Passage, NB	August 8, 1977[+]	5.05	-	F	-	herring weir	Caught in herring weir.	Hogans and Dadswell (1985)	Yes
33	Passamaquoddy Bay off Mascarene Shore, NB	August 8-9, 1977[+]	5.2	-	-	-	herring weir	Trapped in herring weir.	Scott and Scott (1988)	Yes
34	Gulf of St. Lawrence, off Alberton, PEI	August 4, 1983	5.2	-	M	dead	cod gillnet	Caught in cod gillnet; reported by fish harvester. DFO examined, took vertebrae.	Scott and Scott (1988)	Yes
34	Off Tiverton, PEI	July, 1988	4.5	-	-	-	gillnet	Caught in gillnet.	Connors Bros. Ltd. In Mollomo (1998)	Yes
36	Southern Scotian Shelf	November, 1989	-	200	-	dead	pelagic longline (tuna)	Mature. Japanese longliner (bigeye tuna), discarded.	Scotia-Fundy Observer database	Yes
37	Sable Island	Late 1980s	-	-	-	-	-	Tooth recovered from seal carcass.	Campana, pers. comm. (2004)	No
38	65 km west of Sable Island	Oct. 1992		80		dead	cod trawl	Juvenile. Canadian vessel trawling cod, discarded.	Scotia-Fundy Observer database	Yes
39	Bay of Fundy	Early 1990s	4.2	-	-	-	gillnet	Caught in gillnet.	Campana, pers. comm. (2004)	Yes
40	Economy, NS	2010	3.05	-	M	died in weir	herring weir	Juvenile caught in a weir. Reported by fish harvester.	Warren Joyce, pers. comm. (2016)	Yes

	Location	Date	Length (m)	Weight (kg)	Sex	Alive/ Dead	Gear Type	Comments	Reference	Inci. Mor.
41	Economy, NS	August 11, 2011	3.00	-	F	died in weir	herring weir	Juvenile caught in a weir, jaws donated to Natural History Museum. Reported by fish harvester.	Warren Joyce, pers. comm. (2016)	Yes
42	Grand Manan, NB 4°52'93"N, 66°44'32"W	August 17, 2012; 1600 hours	>3.0	-	-	alive	-	Observed preying on a harbour porpoise from Whale Watching boat, *Quoddy Link*.	Turnbull and Dion (2012)	No
43	Passamoquoddy Bay, NB, Sawpit off Swallowtail Lighthouse, Grand Manan 44.7655 Lat -66.7345 Long	August 6, 2013	-	-	-	alive	-	Eating a seal, experienced observer sighted from land. Lat. and Long. are approximate based on location description.	Wong pers. comm. (2016)	No
44	Magdalen Islands	Sept. 25, 2013	-	-	-	-	-	White Shark attack on marine mammal.	Warren Joyce, pers. comm. (2016)	No
45	St. Andrews, NB	July 21, 2014	-	-	-	alive	-	Whale watching with Quoddy Link Marine	Warren Joyce, pers. comm. (2016)	No
46	White Sands, PEI	September 3, 2014	-	-	-	-	-	Dead pilot whale scavenged by white shark.	The Eastern Graphic (2014)	No
47	Parsborro (Minas Passage - West Bay), NS	July 30, 2015	3.66	-	M	dead	unknown	Reported by NS DNR. Carcass washed out to sea, confirmed with teeth.	Warren Joyce, pers. comm. (2016)	Yes
48*	Port Mouton, NS	2015	-	-	-	alive	-	Possibly up to 5 sighted while tuna fishing. Reported by fish harvester.	Warren Joyce, pers. comm. (2016)	No

	Location	Date	Length (m)	Weight (kg)	Sex	Alive/ Dead	Gear Type	Comments	Reference	Inci. Mor.
49*	Bay of Fundy	August 2015	-	-	-	alive	scallop dredge	Reported to DFO fisheries technician by fish harvester.	Warren Joyce, pers. comm. (2016)	No
50*	Seal Island, NS	August 2015	-	-	-	alive	-	Possibly a white attacking a seal, could be a mako. Reported by fish harvester.	Warren Joyce, pers. comm. (2016)	No
51	Alice Head Cove, Saint Margaret's Bay, NS	Late Aug. / early Sept. 2015	5.49	-	-	alive	mackerel / tuna trap	18 foot shark reported by fish harvester.	Warren Joyce, pers. comm. (2016)	No
52	New Brunswick	September 15, 2015	-	-	-	-	-	Porpoise carcass attacked by a white. Reported by John Chisolm.	Warren Joyce, pers. comm. (2016)	No
53	Alma, NB	June 2016	-	-	-	tooth	-	Tooth recovered from lobster buoy.	Warren Joyce, pers. comm. (2016)	No
54	Cape Chignecto Point 45.324444, -64.950386	July 7, 2016	-	-	-	teeth marks	-	Teeth marks in lobster buoy. Photographs taken.	Chisholm pers. comm. (2016)	No
55*	Cross Island, Lunenburg, NS	July 31, 2016	-	-	-	alive	-	Video taken by tourists aboard Lunenburg Ocean Adventures. Shark trying to feed on harbour porpoise. Species unconfirmed.	CTV News: Two Metre Shark Spotted Off Coast of Nova Scotia in Latest Series of Sightings	No
56	St. Andrews, NB	August 1, 2016	5.2	-	-	alive	-	St. Andrews Sport Fishing Co. captured video. John Chisholm confirmed species ID and length.	CBC News: NB Great White Shark Filmed Near St. Andrews, NB	No

	Location	Date	Length (m)	Weight (kg)	Sex	Alive/ Dead	Gear Type	Comments	Reference	Inci. Mor.
57*	LaHave River, Mosher's Island, NS 44.259194, -64.315125.	August 7, 2016	3.05 - 3.66	-	-	alive	-	Sighted by mackerel harvester. Unconfirmed but very likely a white shark based on description.	Warren Joyce, pers. comm. (2016)	No
58*	Delap's Cove, NS (Bay of Fundy)	August 10, 2016	2	-	-	alive	-	One mile past Charlie's Brook, presumed shark observed breaching the surface twice, killing a grey seal.	Warren Joyce, pers. comm. (2016)	No
59	St. Margarets Bay, NS	August 19, 2016	4.6	-	-	alive	mackerel / tuna trap	Caught in mackerel trap and released. Video and photos available.	Warren Joyce, pers. comm. (2016)	No
60*	White Point Shoal, NS 43 56.25' N, 64 43.02' W	September 26, 2016	3.66 - 3.96	-	-	alive	-	Tuna fisherman (rod and reel) reported sighting - no gear interaction. High confidence in species identification.	Warren Joyce, pers. comm. (2016)	No
61*	Approximately 43 55.65' N, 64 44.103' W	Approx Sept 30, 2016	3.05 - 3.66	-	-	alive	-	Tuna fisherman (rod and reel) reported sighting - no gear interaction. High confidence in species identification.	Warren Joyce, pers. comm. (2016)	No

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